Constructed Wetlands for Treatment of Swine Wastewater from an Anaerobic Lagoon

P. G. Hunt, A. A. Szögi, F. J. Humenik, J. M. Rice, T. A. Matheny, K. C. Stone

Abstract. *Animal waste management is a national concern that demands effective and affordable methods of treatment. We investigated constructed wetlands from 1993 through 1997 at a swine production facility in North Carolina for their effectiveness in treatment of swine wastewater from an anaerobic lagoon. We used four wetland cells (3.6 × 33.5 m) with two cells connected in series. The cells were constructed by removing topsoil, sealing cell bottoms with 0.30 m of compacted clay, and covering with 0.25 m of loamy sand topsoil. One set of cells was planted with bulrushes (*Scirpus americanus, Scirpus cyperinus, *and* Scirpus validus*) and rush (*Juncus effusus*). The other set of cells was planted with bur-reed (*Sparganium americanum*)and cattails (*Typha angustifolia *and* Typha latifolia*). Wastewater flow and concentrations were measured at the inlet of the first and second cells and at the exit of the second cell for both the bulrush and cattail wetlands. Nitrogen was effectively removed at mean monthly loading rates of 3 to 40 kg N ha^{-1} day^{-1}; removals were generally >75% when loadings were <25 kg ha^{-1} day^{-1}. In contrast, P was not consistently removed. Neither plant growth nor plant litter/soil accumulation was a major factor in N removal after the loading rates exceeded 10 kg N ha^{-1} day^{-1}. However, the soil–plant–litter matrix was important because it provided carbon and reaction sites for denitrification, the likely major treatment component. Soil Eh (oxidative/reductive potential) values were in the reduced range (<300 mV), and nitrate was generally absent from the wetlands. Furthermore, the wetlands had the capacity to remove more nitrate–N according to denitrification enzyme activity determinations. Our results show that constructed wetlands can be very effective in the removal of N from anaerobic lagoon–treated swine wastewater. However, wetlands will need to be augmented with some form of enhanced P removal to be effective in both P and N treatments at high loading rates.*

Keywords. *Denitrification, Nitrogen, Phosphorus, Plant uptake, Redox potential, Soil accumulation.*

Animal production is a major component of agriculture in the U.S. It is vital to both food stability and economic health. However, there are increasingly visible environmental problems associated with modern animal production. These problems include odors, pathogens, concentrated wastewater, inadequate land treatment sites, residential encroachment, and new regulations. Currently, most swine production enterprises initially treat wastewater in anaerobic lagoons and subsequently apply the treated wastewater to land. This method is satisfactory when large tracts of cropland are available and neighbors are some distance from application areas (Stone et al., 1995). However, these land use and demographic conditions often do not exist. As a result, environmental groups, regulators, and the public are demanding superior treatment alternatives. One of these alternatives is constructed wetlands.

Wetlands have been used successfully for advanced treatment of municipal and residential wastewaters in the U.S. and around the world for over three decades (Hammer, 1989; Kadlec and Knight, 1996). They are generally perceived as a technology that is relatively affordable and operationally simple. Compared to conventional systems, they have less construction, operation, and energy costs plus more flexibility in pollutant loading. They are also flexible in soil specificity; constructed wetlands can be built on aerated upland soils, and the wetland soil conditions will develop when the soils are flooded. These soil conditions will then support aquatic plant life and wetland processes. Their function and reliability for animal wastewater treatment are less documented (Hunt and Poach, 2001).

Generally, the focus of animal wastewater treatment in constructed wetlands is to remove nutrients and thereby decrease the land necessary to receive, transform, and assimilate the remaining nutrients in the wastewater (Knight et al., 2000). Land application is necessary because direct discharge of animal wastewater is not permitted even after treatment. The objectives of this study were to determine: (1) the effectiveness of constructed wetlands in removing N and P from anaerobic lagoon–treated swine wastewater, and (2) the relative importance of treatment components (soils, plants, and microbes) to the functioning of these wetlands.

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Materials and Methods

Study Site

The investigations were conducted from 1993 through 1997 at a swine production facility in Duplin County, North Carolina. The facility was a 2600–pig nursery with pigs that averaged 13 kg. It used a flushing system to recycle wastewater from a single–stage anaerobic lagoon with a volume of 4100 m^3. Residence time of the wastewater in the lagoon was about 120 d. Excess wastewater in the lagoon was applied to land. We used only a portion of the wastewater to conduct the wetland experiment. Irrigation of the wastewater on an adjacent spray field proceeded under normal farm nutrient management protocol.

The constructed wetlands consisted of four 3.6 × 33.5 m wetland cells with two sets of cells connected in series (fig. 1). These dimensions were acceptable to the Natural Resources Conservation Service wetland design needs and the land available for the experiment. They fall within the range described by Kadlec and Knight (1996). The cells were constructed by removing the topsoil, sealing the cell bottoms with 0.30 m of compacted clay, and covering with 0.25 m of loamy sand topsoil. The first cell in each series received inflow wastewater from the lagoon after it was diluted with fresh water to keep the ammonia–N <200 mg L^{-1}. This was accomplished by use of a mixing tank with input from lagoon wastewater and fresh water proportional to the needed dilution. The dilution mitigated possible ammonia toxicity to the wetland plants. The second cell received wastewater from the outflow of the first cell. The effluent from the second cell was collected in a tank and pumped back to the lagoon. This allowed the experiment to be conducted without discharge to streams or impact on the irrigation of wastewater to the spray field.

The first set of cells was planted with bulrushes (*Scirpus americanus, Scirpus cyperinus,* and *Scirpus validus*) and rushes (*Juncus effusus*); they will hereafter be referred to as bulrush wetlands. The second set of cells was planted with bur–reed (*Sparganium americanum*)and cattails (*Typha angustifolia* and *Typha latifolia*); they will hereafter be referred to as cattail wetlands. Plant growth was active during much of the year. Since the water levels in the wetlands were shallow, water temperature closely approximated the air temperature, which had a mean monthly range of 4.2° C to 28.7° C. The median monthly temperature was 19.1° C.

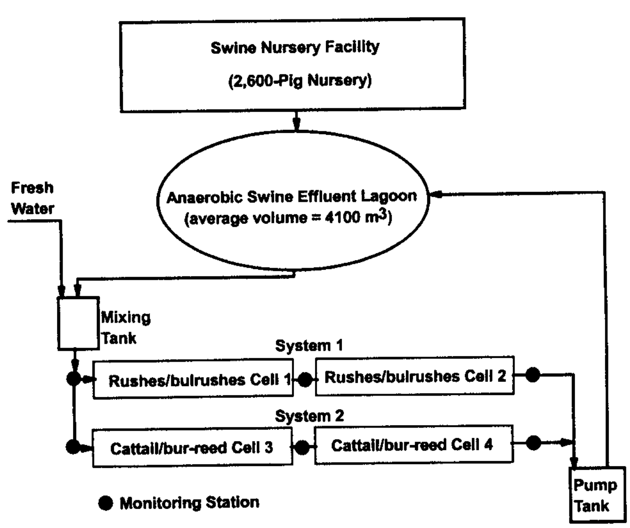

Figure 1. Site schematic of constructed wetlands.

Anaerobic Lagoon Wastewater

Characteristics of the undiluted swine wastewater from the lagoon are indicative of a moderately loaded lagoon (table 1). Wastewater was diluted with groundwater to alter nitrogen (N) and phosphorus (P) loading rates over the five–year period; the mean annual daily loading rates ranged from 4.8 to 27.2 kg ha^{-1} day^{-1} for N, and from 0.9 to 6 kg ha^{-1} day^{-1} for P (table 2). To ensure that we did not overly stress the wetland plants, we started the experiment with low rates and increased them over time. This procedure also allowed us to determine the treatment effectiveness in relation to loading rates. Mean monthly N and P loading rates were more variable than the annual loading rates, ranging from 3 to 40 kg N ha^{-1} day^{-1} and from 1 to 9 kg P ha^{-1} day^{-1}. The N form was primarily ammonia (>90%) for both the inflow wastewater and the effluent; nitrate–N was negligible. The wetland cells were loaded using an automated system with float control valves in the dilution tank. Hydraulic loading rates were determined by flow meters and tipping buckets at the cell inlets and by V–notch weirs using ultrasonic depth sensors (Control Electronics, Morgantown, Pa.) at the outlet of the second cell. The daily hydraulic loading rates ranged from 8 to 11 mm d^{-1} with a mean residency time of 12.5 days per cell (Szögi et al., 2000). Diluted wastewater was applied during June – December in 1993, January – December in 1994, intermittently January – March, and regularly April – December 1995, April – December 1996, and March – November in 1997.

Sampling and Analyses

Water samples were collected at the inlet of the first and second cells and the outlet of the second cell of each wetland system using ISCO automated water samplers (ISCO Corp., Lincoln, Nebr.). The normal protocol was for samples to be

Table 1. Characteristics of non–diluted swine wastewater after anaerobic lagoon treatment.

Parameters	Units	Mean	Std. Dev.
pH	—	7.83	0.14
Total solids	g L^{-1}	1.86	0.47
Volatile solids	g L^{-1}	0.73	0.32
Total organic carbon	mg L^{-1}	235	124
Chemical oxygen demand	mg L^{-1}	737	237
Biochemical oxygen demand	mg L^{-1}	287	92
Total Kjeldahl nitrogen	mg L^{-1}	365	41
Ammonia–nitrogen	mg L^{-1}	347	52
Nitrate–nitrogen	mg L^{-1}	0.04	0.03
Total phosphorus	mg L^{-1}	93	11
Orthophosphate–phosphorus	mg L^{-1}	80	9

Table 2. Mean annual daily loading rates (kg ha^{-1} day^{-1}) of nitrogen and phosphorus into the constructed wetland.

	Bulrush		Cattail	
Year	Nitrogen	Phosphorus	Nitrogen	Phosphorus
1993	4.8	0.9	5.6	1.1
1994	5.6	1.1	6.4	1.3
1995	8.2	1.9	9.1	2.0
1996	11.8	2.3	17.6	3.4
1997	27.2	6.0	14.9	3.2

collected on 8–hour intervals and composited into 3.5–day samples. Two sets of samples were collected: one set in which sulfuric acid was added as a preservative, and one set without acid. Ammonia–N, nitrate–N, orthophosphate–P, total Kjeldahl N (TKN), and total P were determined on the acidified samples using EPA methods (U.S. EPA, 1983). All N and P analyses were conducted with automated analyzers (Technicon Instruments Corp., Tarrytown, N.Y., and Bran+Lubbe Corporation, Buffalo Grove, Ill.). Total N (TN) was the sum of TKN and NO_3–N. Other parameters were determined by *Standard Methods* procedures on non–acidified samples: total solids, volatile solids, total organic carbon, chemical oxygen demand, and biochemical oxygen demand (American Public Health Association, 1992).

Plants (above–ground dry matter) were harvested from 0.25 m^2 in each of three equal sections within a cell for dry matter production and nutrient accumulation by cutting the vegetation at soil level. This protocol resulted in six samples for each wetland system per sampling date. On each section, plots were marked early in the study to minimize disturbance and to avoid re–sampling of the same small plot during a growth season. The disturbance persisted for less than two weeks, and vegetation was recovered by the time of the next sampling date. Plant samples were collected monthly from April through September. They were oven–dried at 65° C to a constant moisture, weighed, ground with a Willey Mill (Arthur H. Thomas, Philadelphia, Pa.) and a Cyclone Mill (UDY Corp., Fort Collins, Colo.). They were then digested using a block digester and analyzed for N and P with an automated analyzer (Gallaher et al., 1976).

Soil aerobic/anaerobic conditions were determined by measuring the soil redox potential (Eh) with a total of 60 platinum electrodes. The electrodes were arranged in sets of five; three sets were installed in each wetland cell with an Ag/AgCl reference electrode for each set. All electrodes were tested for quality control before and after field measurements with quinhydrone in pH 4.0 and 7.0 buffers (Bohn, 1971; Szögi et al., 2000). Electrodes were installed into the soil at 20–, 50–, and 100–mm depths. A data logger (CR7X, Campbell Scientific, Logan, Utah) was used for hourly acquisition of the soil redox potential. Redox potential values were adjusted to standard H electrode potentials (Eh) by adding the potential of the Ag/AgCl electrode (200 mV) to the mV field reading.

Soil samples were collected to a depth of 200 mm yearly (March–April) from three equal sections of each cell. Eight to ten (2.2 cm diameter) samples were taken at random within each section. The undecomposed litter was removed, and the samples were combined into one composite sample per section. They were subsequently air–dried, crushed, sieved at 2 mm, digested using the block digester method, and analyzed for TKN and TP with an automated analyzer.

Denitrification potential was determined using denitrification enzyme activity (DEA) on additional soil samples collected at the 0– to 25–mm depth from four quadrants of each cell every three months for three years. Soil samples were placed in plastic bags, stored on ice, transported to the laboratory, and stored overnight at 4° C. The DEA was measured on the samples using the acetylene reduction method (Tiedje, 1982). In this procedure, soil samples were analyzed for potential denitrification by measure of nitrous oxide production from soil samples incubated at room temperature (~20° C) under four conditions: (1) the control with no additions, (2) addition of the electron acceptor (nitrate), (3) addition of a carbon source (glucose), and (4) addition of both nitrate and glucose. Nitrous oxide concentrations were determined with a Varian Model 3600 CX gas chromatograph equipped with a 15–mCi ^{63}Ni electron capture detector operating at 350° C and a 1.8–m by 2–mm ID stainless steel column containing poropak Q (80–100 mesh).

Data were analyzed by analysis of variance, regression, and least significant difference (LSD) with version 6.12 of Statistical Analysis Systems (SAS Institute Inc, Cary, N.C.).

Results and Discussion

Treatment Effectiveness

Mass N Removal

Substantial removal of N was accomplished over a considerable range of mean monthly loading rates (3 to 40 kg N ha^{-1} day^{-1}) by both the bulrush and cattail wetlands (fig. 2). Regression equations of monthly mean N load versus N

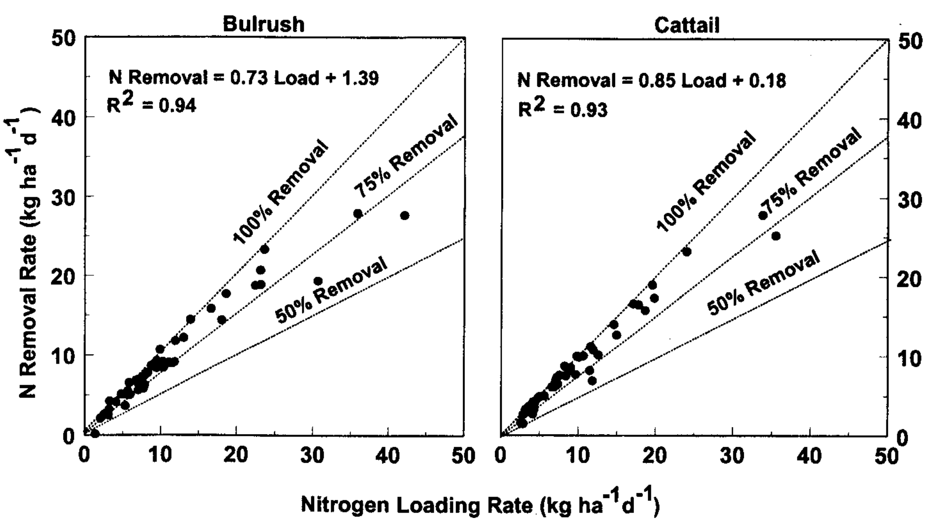

Figure 2. Nitrogen removal rate of constructed wetlands as a function of N loading rates.

removal were: N removal = 0.73 N load + 1.39, $R^2 = 0.94$, and N removal = 0.85 N load + 0.18, $R^2 = 0.93$, for the bulrush and cattail wetlands, respectively. The wetlands were less effective in N removal when loaded at >25 kg N ha^{-1} day^{-1}. However, removals of applied N were always >50% and most were >75%. Furthermore, as shown later in the Denitrification section, the wetlands have much more N removal potential if oxidized wastewater with nitrate N is added.

This potential notwithstanding, the actual N loading rates and treatment efficiencies of this experiment are both very high relative to traditional land application treatment and consistent with other wetland literature. For instance, in Alabama, McCaskey et al. (1994) found 99% to 82% removal of total N from swine lagoon wastewater treated with constructed wetlands that were loaded at 2.5 to 12.5 kg N ha^{-1} day^{-1}. Our results also correspond with those reported by Cathcart et al. (1994) for a marsh–pond–marsh (MPM) constructed wetland system in Mississippi; they obtained mass ammonia–N reductions of 71% when their system was loaded with 14 kg N ha^{-1} day^{-1}. Our wetlands likewise treated N similarly to a MPM wetlands in North Carolina (Reddy et al., 2000). In addition, at loading rates <9.0 kg N ha^{-1} day^{-1}, the treatment of N in our bulrush and cattail wetlands was very similar to that of a saturation–culture–soybean and flooded rice system (Szögi et al., 2000).

After wetlands have dramatically reduced the quantity of N in the wastewater, much less cropland will be required to accept the N load. Moreover, the timing of the applications can be managed more easily to accommodate both weather patterns and crop needs. Each hectare of wetland could remove >7 Mg N each year with three conditions: (1) highest loading rate (40 kg N ha^{-1} day^{-1}), (2) 70% N removal, and (3) 250 days of wetland operation. Fourteen hectares of forage, even at a very high removal rate of 500 kg N ha^{-1} yr^{-1}, would be required to treat the same amount of N as a hectare of wetland. Furthermore, over 45 ha would be required to obtain the same level of N treatment with a row crop such as corn, which would only remove about 150 kg N ha^{-1} yr^{-1}. When this amount of land is not available or expansion of the operation is desired, constructed wetlands can offer a feasible alternative for managing the N load from swine facilities. Additionally, wetland systems are operationally passive, and they cycle N via natural processes.

Despite these advantages, some have cautioned that the high rates of N removal from animal wastewater treatment wetlands could be due to ammonia volatilization (Knight et al., 2000). We thought this loss would not be a major factor because of the relatively neutral pH of the wastewater, the vegetative cover, and the interaction of ammonia gradients and oxygenated micro–sites (Patrick and Reddy, 1976). In a separate investigation, ammonia volatilization from these wetlands was found to be relatively low (generally <15% of the N applied) (Hunt et al., 2000; Poach et al., 2002).

Our N removal results suggest that denitrification is very active in the wetlands. These findings are consistent with those of Harper et al. (2000); they found lower than expected ammonia volatilization but higher than expected denitrification from swine wastewater anaerobic lagoons. Denitrification losses are also supported by recent work on alternative denitrification pathways (Jetten et al., 1999; Luijn et al., 1998). Furthermore, N removals may be increased to much higher levels if the ammonia is nitrified before application to the wetland (Hunt et al., 1999).

N Concentration

Even though our primary focus was on N removal, it was important to know the impact of loading rates on effluent N concentration ([N]) because these data would relate to the strength of the effluent applied to land. Our data showed a moderate correlation between the N load to the first cell and the effluent [N] from both cells 1 and 2 after a natural log conversion (fig. 3). For both the bulrush and cattail wetlands, there was a reduction in [N] via treatment in both cells, and the slopes of the treatment responses for the first and second cells were very similar. In addition, the correlations of ln N load and ln [N] for the bulrush and cattail wetlands were remarkably similar (R^2 = 0.42 to 0.44). As would be expected, the lowest effluent [N] values were obtained with the lowest loading rates. However, even at the higher loading rates, the [N] in the wetland–treated effluent was nearly

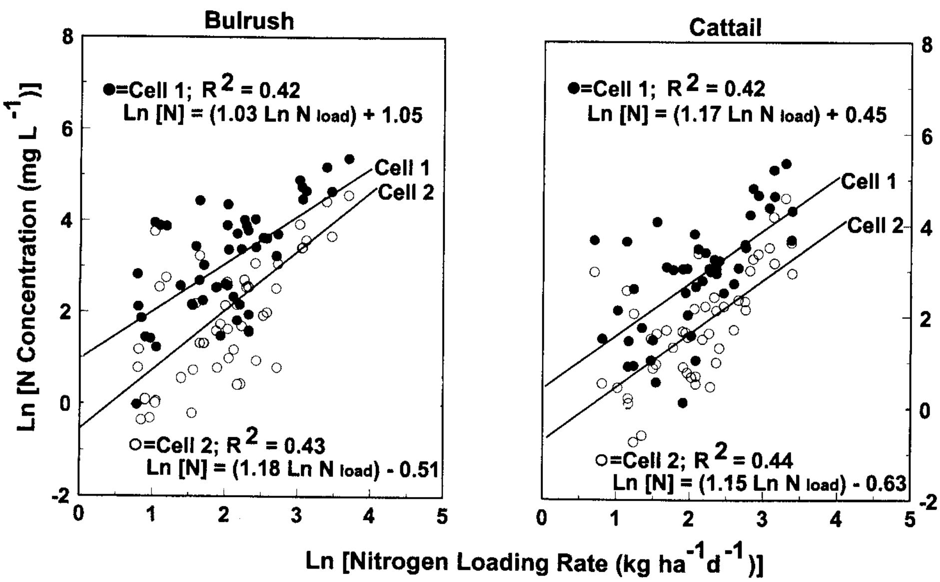

Figure 3. Nitrogen concentration of effluent from constructed wetlands as a function of N loading rates.

always less than that of the lagoon wastewater. Thus, wetland treatment mitigated any potential wastewater strength problems related to its land application.

Mass P Removal

Neither the bulrush nor cattail wetlands were consistently effective in the mass removal of P, and both systems were generally <50% effective when the loading rates exceeded 4 kg P ha^{-1} day^{-1} (fig. 4). There was modest correlation of P load and removal (cattail wetland P removal = 0.50 P load – 0.15, R^2 = 0.48; bulrush wetland P removal = 0.31 P load + 0.33, R^2 = 0.35). The low P removals are consistent with the expectation based on both the reduced Eh conditions of the wetland soil and other reports of P treatment efficiency (Hunt and Poach, 2001; Knight et al., 2000; Szögi et al., 2000).

P Concentration

The correlations of effluent P concentration ([P]) from both cells to P load of cell 1 were moderate after a natural log conversion of the data (fig. 5). At the low loading rates, effluent [P] values were greater for the first cell than the second cell, indicating that treatment of [P] was somewhat similar to that of [N] for both the bulrush and cattail wetlands. However, at the higher loading rates, the regression lines converged, indicating that there was little reduction of [P] after the first cell. This is consistent with somewhat effective treatment at lower loading rates, but declining treatment as loading rates exceeded 4 kg P ha^{-1} day^{-1}. Regression equations for the first and second cells of the bulrush wetlands were, respectively, ln [P] = 1.04 ln P load + 2.10, R^2 = 0.54, and ln [P] = 1.29 ln P load + 1.67, R^2 = 0.50.

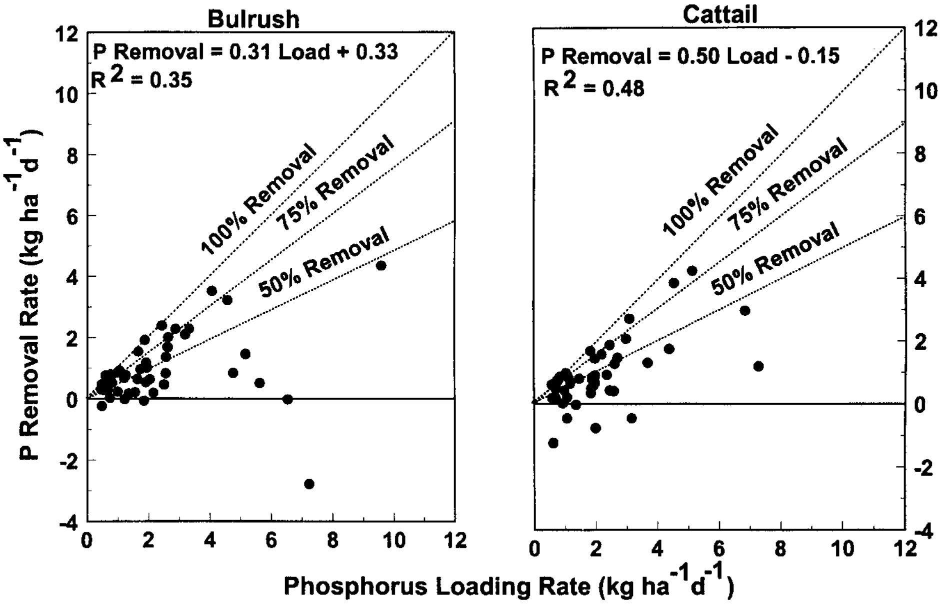

Figure 4. Phosphorus removal rate of constructed wetlands as a function of P loading rates.

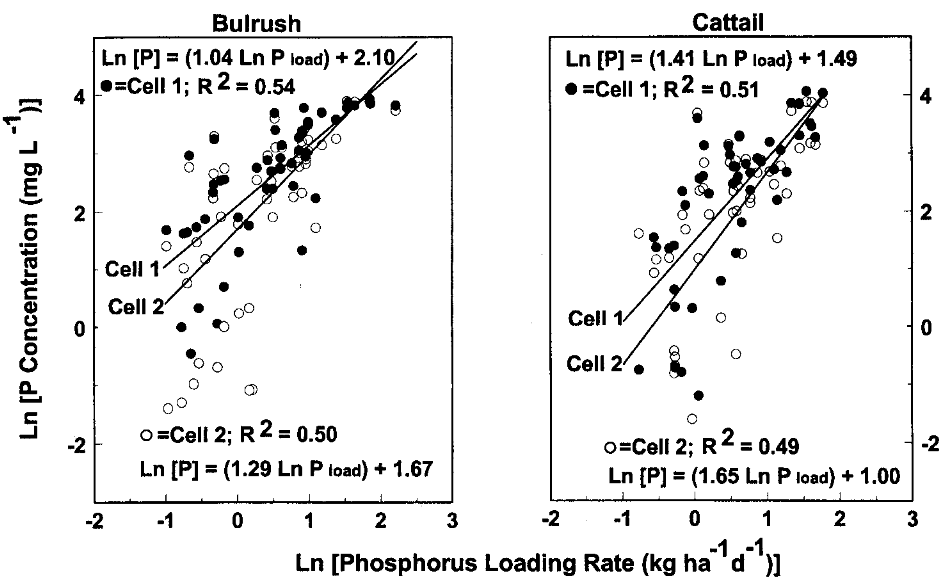

Figure 5. Phosphorus concentration of effluent from constructed wetlands as a function of P loading rates.

Table 3. Plant dry matter, nitrogen, and phosphorus accumulation in constructed wetlands.

Year	Dry Matter ($Mg\ ha^{-1}\ yr^{-1}$)			Nitrogen ($kg\ ha^{-1}\ yr^{-1}$)			Phosphorus ($kg\ ha^{-1}\ yr^{-1}$)		
	Bulrush	Cattail	$LSD_{0.10}$[a]	Bulrush	Cattail	$LSD_{0.10}$[a]	Bulrush	Cattail	$LSD_{0.10}$[a]
1993	21.8	9.9	[a]	360	114	[a]	87	20	[a]
1994	11.6	28.0	[a]	182	359	[a]	37	80	[a]
1995	12.5	11.1	NS	289	222	NS	40	58	NS
1996	25.3	23.8	NS	557	595	NS	163	206	NS
1997	16.3	13.2	NS	384	293	NS	53	53	NS
Mean	17.5	17.2	NS	354	317	NS	76	83	NS

[a] Least significant difference at the 0.10 level.

Regression equations for the first and second cells of the cattail wetlands were, respectively, ln [P] = 1.41 ln P load + 1.49, $R^2 = 0.51$, and ln [P] = 1.65 ln P load + 1.00, $R^2 = 0.49$.

As with N, the reduction of mass P is a greater concern than reduction of effluent [P]. However, in either aspect of P treatment, some form of pre– or post–wetland P removal augmentation is likely needed for a totally functioning wetland treatment system. If a precipitation agent such as alum was added to the wastewater inflow, P would accumulate in the plant litter soil matrix in a relatively stable form (Davies and Cottingham, 1993; Lee et al., 1976). However, the amount of alum needed would be significant, as would be the associated sludge. Alternatively, P could be removed from the wastewater before application to the wetland or from the effluent after wetland treatment (Vanotti et al., 2001a, 2001b); the advantages of one versus another would depend on the environmental restrictions and operational goals.

Treatment Components

Plant Growth and Nutrient Accumulations

During the five years of this investigation, there was a substantial range of annual plant dry matter accumulation (from 11 to 28 $Mg\ ha^{-1}\ yr^{-1}$), as shown in table 3. Bulrushes and cattails alternate in which had the highest dry matter ($LSD_{0.10}$) in 1993 and 1994. Causes of this large range included yearly variable community composition and insect and disease pressure. However, the five–year means were very similar for the bulrush and cattail wetlands (17.5 and 17.2 $Mg\ ha^{-1}$, respectively). Moreover, these means are consistent with other wetland systems (DeBusk and Ryther, 1987).

Plant N annual accumulations ranged from 114 to 595 $kg\ ha^{-1}\ yr^{-1}$. However, the mean annual accumulations were not significantly different for the bulrush and cattail wetlands (354 and 317 $kg\ ha^{-1}\ yr^{-1}$, respectively). Plant P annual accumulations ranged from 20 to 206 $kg\ ha^{-1}\ yr^{-1}$. As with N, the mean annual accumulations of P were not significantly different for the bulrush and cattail wetlands (76 and 83 $kg\ ha^{-1}\ yr^{-1}$, respectively). At the low loading rates, the N and P annually accumulated by the plants were significant components of the wetland's annual nutrient budget (~30% and 38%, respectively). However, once the application rate exceeded 10 $kg\ N\ ha^{-1}\ day^{-1}$, plant accumulation of N and P was a minor component (<3%).

Nonetheless, the accumulation of plant dry matter and uptake of nutrients are very important. The uptake of nutrients provided for nutrient storage via internal cycling in the wetland. Since plant dry matter was not harvested, it accumulated on the wetland bottom after the plants had aged and their aerial parts had succumbed to frost. This accumulation allowed a significant litter layer to become established and function as both a source of carbon and an extensive reaction surface for microorganisms. In particular, the carbon exuded from the roots along with the carbon in the dead plant litter provided the energy necessary to drive the denitrification process. This may be particularly critical if high rates of nitrified wastewater were to be added. Hunt et al. (1999) reported the advantage of plant litter for denitrification when ~50 $kg\ N\ ha^{-1}\ day^{-1}$ were added to wetland microcosms.

Redox Conditions

The redox state (Eh) of the wetland soil was generally below the level at which nitrate is stable (<300 mv, table 4). Additionally, the O_2 in the water was usually very low (<2% saturation). The soil in wetland cells with bulrushes had higher Eh values (more oxidized) than the soil dominated by cattails. This is consistent with their relative oxygen transport capacities (Reddy et al., 1989). Since wetland plants can transport O_2 from leaves and stems to roots, they can provide oxidized microenvironments in the anaerobic root zone (Armstrong, 1964; Kadlec and Knight, 1996). The juxtaposition of aerobic and anaerobic zones at the root–water–soil interface is critical to the treatment of wastewater (Good and Patrick, 1987; Hunt and Lee, 1975). Thus, the efficient use of wetlands for wastewater treatment can be highly affected by the O_2 transport capacity of plant–root systems and O_2 diffusion across the air–water interface. Oxygen availability is also affected by the O_2 demand of the wetland. These Eh values reflect both the low oxygen conditions of a flooded soil system and the high oxygen consumption during microbial degradation of the wastewater biochemical oxygen demand (BOD) along with plant exudates and litter. However, it must be noted that these Eh values were obtained from the soil matrix. The Eh may be higher at the liquid–air interface or at the root–liquid interface than could be measured by our Eh electrodes because they were too large to detect the microenvironment differences. Nonetheless, these values document the strongly reduced Eh conditions of the wetland soil matrix and are consistent with the general absence of nitrate–N in soil pore water samples (Szögi and Hunt, 2000).

Table 4. Mean soil redox (Eh) values (mv) for wetland cells used for treatment of swine lagoon wastewater as influenced by the predominant plant community.

Cell Position	Bulrush		Cattail	
	Mean	Std. Dev.	Mean	Std. Dev.
1st	130	179	105	145
2nd	308	285	196	228

Table 5. Applied and soil accumulated nitrogen and phosphorus in constructed wetlands.

Nutrient	Year of Study	Bulrush Applied (kg ha^{-1})	Bulrush Accumulated[a] (kg ha^{-1})	Cattail Applied (kg ha^{-1})	Cattail Accumulated (kg ha^{-1})	Wetland Differences $LSD_{0.10}$
Nitrogen	1	1043	245	1217	201	NS
	2	3102	449	3535	443	NS
	3	6111	542	6852	397	NS
	4	10427	1081	13260	976	NS
	5	19144	1057	18035	997	NS
$LSD_{0.10}$[b]			161		82	
Phosphorus	1	202	245	243	264	NS
	2	602	143	704	−19	[b]
	3	1276	127	1443	−2	[b]
	4	2118	35	2695	−14	NS
	5	4043	250	3719	53	[b]
$LSD_{0.10}$			82		35	

[a] The accumulated values are the increase or decrease from the initial soil N and P content values of 453 and 551 kg ha^{-1}, respectively.
[b] Significant by least significant difference at the 0.10 level.

Soil Accumulations of N and P

Physical and chemical processes of the wetland promoted N and P accumulations in both the litter layer and mineral soil. Thus, we anticipated soil accumulations of N and P (Szögi and Hunt, 2000; Szögi et al., 2000). Mean N accumulation for the wetland systems reached 1027 kg ha^{-1} during the final year (table 5), yet this accumulation was relatively small (<10%) compared to the greater than 18 Mg of N applied during the study period. Mean N accumulations were not significantly different at the 0.10 level between the bulrush and cattail wetlands in any year.

However, the spatial patterns of N accumulation in the bulrush and cattail wetlands were strikingly different. In the bulrushes, mean soil N accumulation decreased with distance from the cell 1 wetland inlet (distance from inlet = DI) to the outlet of cell 2 (accumulated soil N = 848.4 − 5.18(DI) kg ha^{-1}, $R^2 = 0.95$), as shown in figure 6. This was not true for the cattail wetlands ($R^2 = 0.25$). This difference in soil accumulation may have been related to the higher oxidative status in the bulrush wetlands, which could have promoted more nitrate formation and consequently more denitrification.

When these relatively small soil N accumulations are considered in terms of the total N balance, major N gaseous losses can be inferred. To summarize our accounting for the fate of the applied N, we found relatively little N: (1) left the wetland in the effluent, (2) leached through the clay liner, or (3) accumulated in the plant–soil matrix. As discussed earlier and in the subsequent section, the loss mechanism is most likely denitrification.

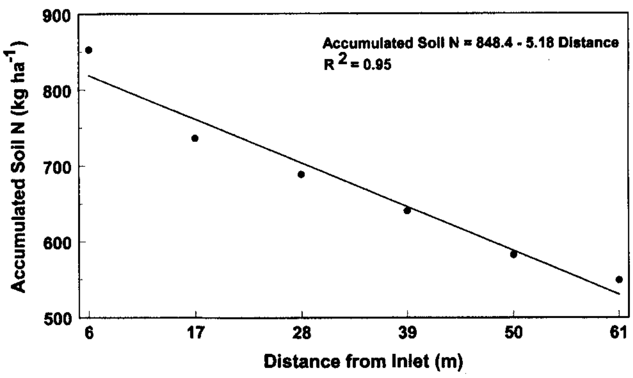

Figure 6. Soil nitrogen accumulation from swine wastewater as a function of distance from constructed wetlands inlet of the bulrush wetlands.

The P did not accumulate in the wetlands over the study period (table 5). In fact, it actually decreased in the cattail wetland. Correlations of P accumulation and distance from the wetland inlet were low ($R^2 < 0.17$). These findings are consistent with the general reduced Eh conditions of the wetland soils, especially in the cattail wetlands. In contrast to N, P was present in the effluent in significant quantities. This was an expected result because these wetlands were loaded with large amounts of P and the reductive environment could promote P solubility. As discussed earlier, when P loads are high, some form of treatment augmentation will be necessary to obtain low levels of P in the effluent.

Denitrification

Since denitrification was indicated as the predominate means of N removal, it was important to understand the likely limiting factors for denitrification. These limits could be lack of sufficient (1) denitrifying microbial microorganisms, (2) nitrate as an electron acceptor, or (3) carbon as an energy source to drive the microbial respiration necessary for denitrification. The denitrification potential was higher in the control sample from bulrush than from cattail wetlands (table 6). This is likely due to the establishment of better nitrifying and denitrifying microbial populations in the bulrush wetlands, where the oxidative conditions would have allowed better production of nitrates as well as subsequent denitrification in the predominating anaerobic soil environment. Addition of nitrate significantly increased denitrification for both the bulrush and cattail wetlands (203% and 282%, respectively). Addition of glucose did not cause a significant increase in denitrification in either wetland system. Addition of both nitrate and glucose produced the highest level of denitrification in both systems, but the increase was not significantly greater than that obtained with only the addition of nitrate. Consequently, we deduced that nitrate was the limiting factor for denitrification based on the denitrification enzyme activity.

Table 6. Denitrification potential (μg N g soil^{-1} hour^{-1}) for constructed wetlands as determined by denitrification enzyme activity of soil samples from the 0 to 25 mm depth.

Amendment	Bulrush	Cattail
Control	0.98	0.47
Nitrate added	1.99	1.33
Glucose – C added	1.40	0.31
Nitrate + C added	2.33	1.52
$LSD_{0.10}$[a]	0.84	0.57

[a] Least significant difference at the 0.10 level.

This finding is consistent with both the literature (Reed, 1993) and our postulation that denitrification would be limited by nitrate availability in these soil samples, which had generally ideal conditions for denitrification. These important conditions include: (1) the Eh of the soil was consistently in the reduced Eh range, indicating low levels of O_2, (2) substantial populations of anaerobic microorganisms were present, and (3) large amounts of carbon were available in the wastewater, plant root exudates, and plant litter. Additionally, these findings are consistent with those of a separate study at this site, where we found very high levels of N removal when nitrified wastewater was added to wetland microcosms at loading rates >40 kg ha^{-1} day^{-1} (Hunt et al., 1999).

Conclusions

Both the bulrush and cattail wetlands were effective in removing N (>50%) at monthly loading rates as high as 40 kg N ha^{-1} day^{-1}. At these rates, wetlands would be more than 40 times as effective per unit area as a typical row crop for assimilation and transformation of applied N.

Wetland plant and soil accumulations of N were a small portion of the N removed. Denitrification appears to be the predominant removal process.

The redox conditions of the wetland soil were in the reduced Eh range where neither oxygen nor nitrate would be stable. These conditions were consistent with our determination that denitrification was limited by nitrate availability. They were also consistent with the limited removal of P, particularly at loading rates >4 kg P ha^{-1} day^{-1}.

Constructed wetlands offer substantial benefits via their characteristics of passive operation and natural processes, yet their treatment effectiveness for swine wastewater from anaerobic lagoons could likely be enhanced by pre–wetland nitrification and phosphorus precipitation. Moreover, as with other technologies, wetlands are likely to work best when used as part of a total waste management system.

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